DECLINING ICE DURATION AND CHANGING ICE QUALITY DRIVE KEY UNDER-ICE ECOLOGICAL CHANGES IN WISCONSIN LAKES

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Abstract

Lakes are rapidly losing ice cover. We expect that ice loss will have widespread ecological consequences, but these consequences are poorly understood, particularly over long temporal scales. Here, we ask: how do changing ice duration and ice quality influence under-ice photic depth, water temperatures, dissolved oxygen concentrations, and chlorophyll *a* concentrations? We analyzed winter data from 11 lakes across Wisconsin, USA, with time-series extending from 1982 to 2023. Shorter ice durations and less snow were associated with deeper photic depths, colder under-ice water temperatures, and higher dissolved oxygen saturations. Snow and ice conditions were significantly related to chlorophyll *a* concentrations; however, the relationship was mediated by nutrient concentrations. We suggest that the drivers of under-ice ecology are as complex as those of the open water season and advocate for the use of integrative models to understand the future of ice-covered lakes.

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Figure 5. Partial dependence of response variables with key ice and snow variables. The first and second most important ice and snow variables within each model were selected. Plots represent the partial dependence of: a) photic depth on fall snow depth; b) photic depth on ice duration; c) bottom water temperature on ice-on day of year; d) surface water temperature on seasonal average snow depth; e) bottom dissolved oxygen saturation on seasonal average snow depth; f) surface dissolved oxygen saturation on mean white ice thickness; g) log chlorophyll *a* concentration on in situ snow depth; and h) log chlorophyll *a* concentration on mean white ice thickness. Dashed lines represent models with low fit or variables with low importance. Ticks on the x-axis indicate intervals each containing 10% of training observations.

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Phoenix Sandrock conceptualized the manuscript, developed the methodology, acquired and cleaned data from the cited repository, performed all formal analyses, created all figures, assisted with data validation, and wrote the manuscript.

Joshua Culpepper assisted with conceptualizing the manuscript, provided feedback and guidance on figures, and assisted significantly with the development and revision of the manuscript.

Arnab Shuvo provided feedback and guidance on the data analysis, including development of the methodology, data acquisition and cleaning; validated the analysis, and reviewed the manuscript.

Sapna Sharma supervised and provided guidance for all aspects of the project, as well as provided the funding for the project.

Chapter One: Introduction

1. 1 Background

Lakes are losing ice rapidly in response to climate change (Huang et al. 2022). Ice duration is shorter, with average declines of approximately 7.2 days per decade from 1992-2016 in Northern Hemisphere lakes (Sharma et al., 2021), and projected to decrease by another 10-28 days by late century under continuing climate warming (Basu et al., 2024). Lake ice is also becoming thinner. For example, between 1961 and 2017, ice thickness declined in lakes across Northern Poland by 2.5 cm per decade on average (Bartosiewicz et al., 2021). Moreover, ice quality (the ratio of black to white ice thickness) may be changing, but trends are more challenging to ascertain owing to limited records. The few lakes with long-term ice quality records suggest an increase in the ratio of white to black ice in lakes, as well as declining overall ice thickness (Culpepper et al., 2024). Given that 86% of lakes worldwide freeze (Korver et al., 2024), these changes in ice cover will have widespread cultural, economic, and ecological implications for the majority of lakes (Cavaliere et al., 2021; Filazzola et al., 2024; Hampton et al., 2024; Knoll et al., 2019).

Changes in the duration and quality of ice have diverse and complex effects on the ecology of lake ecosystems (Hampton et al., 2024). Ice is considered a “master variable” that governs the behaviour of lake ecosystems under ice (Ozersky et al., 2021). Ice can influence the ecology of lakes by creating a physical barrier between the water column, the atmosphere, and the watershed, slowing or preventing the exchange of matter and energy (Jansen et al., 2021). For example, ice cover is an important driver of under-ice water thermal profiles (Kirillin et al., 2012) and inverse stratification in lakes, a process which may be disrupted under climate warming (Austin et al., 2025, Woolway et al., 2021). A few studies have found that delayed ice-on can result in colder under-ice water temperatures, as delays in ice-on allow the lake to shed

heat into the atmosphere for longer periods (Olesky & Richardson 2024; Yang et al., 2021). Moreover, lake ice can block a substantial portion of the light entering the water column (Bolsenga 1981), leading to shallower photic depths. The potential of lake ice to block light depends on its quality; black ice can transmit up to 95% of light (Bolsenga & Vanderploeg 1992), whereas white ice transmits about 20% (Maeda & Ichimura, 1973). Snow can reduce light transmission substantially, with just 5 cm of snow blocking 80-90% of available light (Jakkila et al., 2009).

The relationship between light availability and ice cover has important implications for under-ice productivity. Productivity under ice is generally lower than that during the open-water season (Hampton et al., 2017). Under-ice mean chlorophyll a concentrations are approximately 43% of their summer concentrations, and under-ice phytoplankton biovolume is about 16% of summer biovolume (Hampton et al., 2017). However, the relationship between ice duration and productivity is less clear. Studies on two or three winters in lakes have found both increasing algal biomass under declining ice duration (Dokulil & Herzig 2009) and maximum algal productivity during years with average ice duration (ex. Scordo et al., 2021, Yang et al., 2016). Snow on ice complicates this relationship as the high albedo of snow blocks light (Jakkila et al., 2009). For example, Hrycik and Stockwell (2021) found that phytoplankton biomass was lower in light-limited conditions in an under-ice mesocosm experiment. In another mesocosm experiment, chlorophyll a concentrations under ice were higher when snow was removed from the ice (Knoll et al., 2023). Furthermore, Tapics et al. (2024) found that, within a single ice-on season, phytoplankton biomass increased when snow depth declined and light availability increased. Despite recent interest in the effect of snow and ice quality on under-ice productivity, existing studies are typically conducted within a single year and often use mesocosms. To our

knowledge, there have not yet been any long-term studies quantifying the influence of snow and ice conditions on under-ice productivity.

Ice also influences under-ice oxygen dynamics. Lake ice largely prevents gas exchange between the water column and the atmosphere. This can lead to severe oxygen depletion in lakes, which has resulted in mass mortality events of fish and animals (Fang & Stefan, 2000; Greenbank, 1945). In shallow lakes, under-ice algal blooms can be followed by rapid oxygen depletion as blooms decay (Zhang et al., 2024). A study on high-altitude systems found that shorter ice duration leads to higher dissolved oxygen concentrations under ice, as longer ice durations can create and maintain persistent hypoxic conditions (Smits et al., 2021). Flaim et al. (2020) found in another mountain lake that later ice-on allowed for longer fall mixing, increasing hypolimnetic oxygen both under ice and throughout the year. Ice-covered lakes can also become supersaturated with oxygen, as oxygen produced during photosynthesis or other processes is not released into the atmosphere (ex. Phillips & Fawley 2002). In addition, the responses of lakes to declining ice duration depend on their morphology, with larger lakes becoming more oxygenated under ice as the season shortens, but small lakes becoming more prone to hypoxia and anoxia (Jansen et al., 2025).

1.2 Motivation and research questions

There are still gaps in our understanding of how changing ice duration and ice quality will affect under-ice ecology. We use a unique data set of 11 Wisconsin lakes with up to 40 years of continuous seasonally resolved data to investigate the role of ice conditions on under-ice ecology. Specifically, we ask: 1) How is ice duration changing in a set of diverse lakes in Wisconsin? 2) How is ice quality changing in these lakes? 3) How are changing ice duration and ice quality influencing key under-ice ecological parameters, including a) photic depths, b) water

temperatures, c) dissolved oxygen concentrations, and d) chlorophyll a concentrations? Studies with long time-series are critical in understanding lake ecological responses to ice loss and changing ice quality, given accelerating ice loss (Sharma et al., 2021). Our study uses 40 years of data to explore how key ecosystem variables respond to changing ice duration and ice quality, providing a first look at long-term patterns of the impacts of climate change and ice conditions on under-ice ecology.

CHAPTER TWO: METHODS

2.1 Data acquisition

We acquired data from the core study lakes of the North Temperate Lakes (NTL)-Long-Term Ecological Research (LTER) program, a set of 11 lakes in Wisconsin, USA, with diverse morphologies and trophic statuses (Magnuson et al., 2024; Figure 1, Table S1). Seven of the lakes are located in northern Wisconsin (Allequash, Big Muskellunge, Crystal, Sparkling, and Trout Lakes, as well as Crystal Bog and Trout Bog) and four of the lakes are found in southern Wisconsin (Fish Lake, Lake Mendota, Lake Monona, and Lake Wingra).

The geology of the sites is defined by glacial deposition during the Quaternary period (Rose et al., 2025). The northern lakes are located in low-relief outwash plains and fans composed of stratified sandy gravel, gravelly sand, sand, and silt deposited by meltwater flows (Rose et al., 2025). The southern lakes are in lake plains formed by ice-marginal lakes, and sit on stratified and well-sorted sandy gravel, sand, silt, and clay (Rose et al., 2025). Both the northern and southern lakes are near drumlins and streamlined landforms (Rose et al., 2025). The lakes have diverse morphologies (Table S1), physics, chemistry, and biology (Table S2).

In northern Wisconsin, in-situ measurements of lake parameters (ice, physics, chemistry, and biology) began in 1982 and are sampled 2-3 times per ice-on season. In southern Wisconsin, in-situ winter measurements began in 1996 and occur once per ice-on season. We calculated averages of all sampling events within the winter season for our variables of interest to calculate one mean value per lake per year.

Ice duration data for Lake Mendota and Lake Monona are available from 1856 to 2024, and from 1983 to 2024 for the nine northern lakes, with one annual ice duration measurement. Fish Lake and Lake Wingra were excluded from the ice duration analysis owing to a relatively

short ice duration record. In-situ measurements of ice quality include snow depth, total ice thickness, black ice thickness, and white ice thickness.

Light extinction in the lakes was measured using a light meter lowered through the water column on a boom arm to keep it below the ice sheet. We defined photic depth as the point where the light in the water column dropped below 1% of surface light (depth in metres). In instances where the light at the ice-water interface was already less than 1% of surface light, we considered the photic depth to be at the ice-water interface and assigned it a value of 0m. We also acquired information on surface and bottom water temperature (°C) and dissolved oxygen (% saturation). The surface value for water temperature was defined as the water temperature at 1m depth, whereas the surface value for dissolved oxygen was taken at the ice-water interface. Both bottom water values were taken at the maximum measured depth at the auger hole. We calculated the average chlorophyll *a* (µg/L), total nitrogen (µg/L), total phosphorus (µg/L), and dissolved organic carbon (mg/L) concentrations in the photic zone.

We compiled meteorological data from ERA5-Land, a climate reanalysis product from Copernicus (Muñoz-Sabater et al., 2021). The meteorological variables we acquired from ERA5-Land were: air temperature at 2m above the Earth's surface (°C); daily average precipitation (mm); daily average snow depth (mm); and downward shortwave radiation (W/m^2). ERA5-Land has an hourly temporal resolution and a 0.1° x 0.1° spatial resolution. We obtained these data through Climate Engine (Huntington et al., 2017) and produced averages for four time periods: fall (September, October, November), winter (December, January, February), spring (March, April, May), as well as an ice cover season average (November to March). Note that, due to their proximity, some lakes are within the same grid cell of ERA5-Land and thus have identical values within a season.

2.2 Data analysis

2.2.1 Ice duration

All statistical analyses were conducted in the R language programming environment (R Core Team, 2024). First, we used Mann-Kendall tests and Sen's slopes to calculate trends in ice duration and ice quality over time (packages:"Kendall", McLeod 2022; "trend", Pohlert 2023). Next, we calculated the ice duration anomaly in each lake and year by subtracting the annual ice duration from the overall average ice duration of that lake over the complete time-series (as in Basu et al., 2024). We then averaged the ice duration anomaly across lakes within each year to produce an average anomaly value per year. Due to the differing length of their series, we analyzed the northern and southern groups of lakes separately; we did not quantify ice duration trends in Fish Lake and Lake Wingra because of the short time-series.

2.2.2 Ice quality

We also used Mann-Kendall tests and Sen's slopes to investigate changing ice quality. We calculated the average thickness of total ice, black ice, and white ice across lakes within a season to produce a single point per year for each ice variable. Ice quality data were available from 1983 to 2023 for the northern lakes. In the southern lakes, total ice thickness data were available from 1996 to 2023 for Lakes Mendota and Monona, and from 1997 to 2023 for Fish Lake and Lake Wingra. Ice quality measurements for all southern lakes were available from 2001 to 2023.

2.2.3 Photic depth, temperature, dissolved oxygen, and chlorophyll a

We aimed to understand the effect of ice conditions on under-ice photic depth, water temperature, dissolved oxygen saturation, and chlorophyll *a* concentrations. Thus, we built models with each of these four variables as response variables. Our predictor variables included

ice conditions, meteorological conditions, lake morphological characteristics, and under-ice water quality conditions. Additionally, chlorophyll *a* concentration was used as a predictor in the model for photic depth, in addition to photic depth, water temperature (surface and bottom), and dissolved oxygen saturation (surface and bottom) being used as predictors in the model for surface chlorophyll *a* concentration. Many of our predictor variables violated assumptions of normality. Thus, we log-transformed total nitrogen, total phosphorus, and chlorophyll *a* concentration data prior to analysis.

We used random forest modeling (R package: "caret", Kuhn, 2008), which is a tree-based approach that effectively account for the nonlinearity in the relationships between our predictor and response variables, as well as frequent collinearity and interaction amongst predictor variables (De'ath & Fabricius 2000; Breiman 2001). For each response variable, we constructed a random forest model with 500 regression trees. We allowed the model to test between two and nine variables at each split using ten-fold cross-validation and selected the model that explained the most variation in the response variable. For water temperature and dissolved oxygen concentration, we produced separate random forest models for surface and bottom values, owing to their physical separation during inverse stratification. Each model initially included all predictor variables that had the potential to influence each response variable and was constructed using observations of that variable with no missing values. We used a backward selection procedure (as in Kuhn et al., 2008) to sequentially remove variables with a low relative importance based on the "importance" function of the "randomForest" package (Liaw & Wiener, 2002) with 50-fold cross-validation. We assessed model fit using the model's R^2 value and used partial dependence plots (R package: "pdp", Greenwell, 2017) and the relative variable

importance within the random forest model to determine how the most influential ice and snow variables affected each response variable.

CHAPTER THREE: RESULTS

3.1 Ice duration

Ice duration decreased by an average of 1.2 days per decade in the seven northern Wisconsin lakes from 1982 to 2024 (Figure 2a). Allequash Lake had the most pronounced ice loss at 2 days per decade, while Trout Bog had the slowest decline at 0.5 days per decade. All of the northern lakes showed declining trends in ice duration, albeit insignificantly over the 42-year time-series (Table 1). Lakes Mendota and Monona showed significant declines in ice duration of 1.8 days per decade and 2.0 days per decade respectively between 1856 and 2024 (Table 1, Figure 2b).

3.2 Ice quality

Changes in ice quality and thickness were complex and varied between the northern and southern Wisconsin lakes (Figure 3).

3.2.1 Total ice thickness

From 1982 to 2023, the trend for total ice thickness in the northern lakes was -0.3 cm per decade on average, ranging from -0.15 cm per decade in Big Muskellunge Lake to 1.5 cm per decade in Crystal Bog. In southern Wisconsin, the average trend for total ice thickness was 2.2 cm per decade from 1996 to 2023, but ranged from 4.7 cm per decade in Lake Monona to -0.3 cm per decade in Lake Mendota (Table 1). Although none of the ice thickness trends were statistically significant, they exemplify the variation in ice thickness that can occur within a small geographic area experiencing similar climate drivers. This variation is particularly pronounced in Lakes Mendota and Monona, which had opposing trends despite their similar size and proximity.

3.2.2 Black ice thickness

From 1982 to 2023, northern lakes exhibited decreases in black ice thickness at 0.4 cm per decade (Table 1, Figure 3B). Trends in black ice thickness ranged from a decline of 2.3 cm per decade in Sparkling Lake to an increase of 0.9 cm per decade in Trout Lake (Table 1). The only significant trend in black ice thickness was a decline of two centimetres per decade in Trout Bog (Table 1). In contrast, black ice thickness increased in most of the southern lakes between 2001 and 2023, with an average increase of 1.5 cm per decade (Table 1). Black ice trends in the Southern lakes ranged from 4.4 cm per decade in Lake Wingra to -1 cm per decade in Lake Mendota (Table 1) but were not significant, likely due to the limited 22-year time series.

3.2.3 White ice

On average, white ice thickness increased in the northern lakes by 1.0 cm per decade between 1982 and 2023, whereas white ice thickness declined in the southern lakes by 1.2 cm per decade between 2001 and 2023 (Figure 3; Table 1). Northern lakes showed smaller variation in white ice trends, ranging from -1.4 cm per decade to 2.0 cm per decade (Table 1). However, southern lakes exhibited greater variation in white ice thickness trends, ranging from -4.1 cm per decade to 2.0 cm per decade. The greater variability in trends in the southern lakes may be attributable to their restricted time series. Only two lakes showed significant trends in white ice thickness: Trout Bog in northern Wisconsin (2.0 cm per decade) and Lake Wingra in southern Wisconsin (-4.1 cm per decade; Table 1).

3.2.4 Snow depth

The northern and southern lakes showed divergent trends in snow depth. Snow depth increased across the northern lakes by an average of 1.1 cm per decade between 1982 and 2023. Snow depth trends in northern lakes ranged from 0.7 cm per decade in Lake Allequash to 1.6 cm

per decade in Crystal Lake (Table 1). Increases in snow depth were unanimous, but were only significant in Crystal Lake (1.6 cm per decade) and Crystal Bog (1.1 cm per decade; Table 1). While snow depth increased across the northern lakes, southern lakes experienced a decline in snow depth at a rate of 1.4 cm per decade between 2001 and 2023 (Table 1). Decreases in snow depth in the southern lakes ranged from 0.4 cm per decade in Lake Mendota to 2.1 cm per decade in Lake Monona (Table 1). While none of these trends were significant, all southern lakes showed declining snow depth over the study period (Table 1).

3.3 Under-ice photic depth

The random forest model for under-ice photic depth explained 62% of the variation based on a combination of meteorological, lake morphological, chemical, snow and ice, and biological characteristics (Figure 4a). The three most important explanatory variables influencing under-ice photic depth were lake mean depth, chlorophyll *a* concentration, and fall snow depth. Under-ice photic depth was lowest in shallow lakes with high chlorophyll *a* concentrations, typically bogs with high DOC (Figure 4a). Fall snow depth was the most important ice and snow variable in our model and was ranked third out of ten variables. Less snow accumulation in fall caused a rapid increase in under-ice photic depths ($r = -0.37$). In these Wisconsin lakes, under-ice photic depths were much deeper when less than 10 mm of snow accumulated in fall (Figure 5a). Additionally, shorter ice duration ($r = -0.52$) and less snow measured in-situ ($r = -0.21$) led to deeper photic depths. The remaining driver in the model was shortwave radiation, with higher shortwave radiation driving deeper photic depths (Figure 4a).

3.4 Under-ice water temperature

The random forest model for under-ice bottom water temperature explained 70% of the variation; lake area and lake mean depth were the most important drivers (Figure 4c). Under-ice

water temperatures were coldest in large, deep lakes. Fall meteorology was also an important driver of under-ice water temperatures, with fall snow depth, shortwave radiation, precipitation, and air temperature all featuring in the final model (Figure 4c). Moreover, later ice-on was an important driver of colder under-ice bottom water temperatures ($r = -0.32$, rank 3/14) (Figure 5c). Under-ice bottom water temperatures were warmest when ice-on was before December, and were cooler when lakes froze in or after early January (Figure 5c).

The random forest model for under-ice surface water temperature explained 53% of the variation and included meteorological, lake morphological, and snow and ice parameters. The most important variable was lake area, followed by seasonal precipitation and lake mean depth (Figure 4b). Under-ice surface water temperatures were coldest in large, deep lakes, and in lakes with higher precipitation. Snow depth also emerged as a critical driver ($r = -0.25$, rank = 4/12), with less in-situ snow resulting in warmer under-ice surface water temperatures (Figure 5d). While the model for bottom water temperature selected all possible fall variables, the model for surface water temperature selected a mix of fall, winter, spring, and seasonal meteorology (Figure 4b).

3.5 Under-ice dissolved oxygen saturation

The random forest model for under-ice bottom dissolved oxygen saturation explained 47% of the variation. High DOC concentrations were the main predictor of hypoxia in bottom waters, and were mostly observed in the two bog lakes. Bottom water dissolved oxygen saturations were also lower with higher total nitrogen concentrations (Figure 4d). Additionally, under-ice bottom dissolved oxygen concentrations were higher when seasonal snow depth was less than 30 mm ($r = -0.17$, rank = 3/13), above which dissolved oxygen saturation declined rapidly (Figure 5e).

Surface dissolved oxygen saturation was more predictable than bottom dissolved oxygen saturation, with our random forest model explaining 85% of the variation. The most important variables were DOC, lake area, and lake mean depth, suggesting that lake morphology was more important for surface dissolved oxygen than for bottom dissolved oxygen (Figure 4e). Surface dissolved oxygen saturation was lowest in bog lakes, which had high DOC and a small surface area. However, high saturation in surface waters was also common. Of the 298 available dissolved oxygen measurements, 101 were supersaturated. Supersaturation primarily occurred in eutrophic Lake Mendota and Lake Monona, or Sparkling Lake and Crystal Lake, which are clear. Lastly, thicker white ice reduced surface dissolved oxygen saturation (rank = 4/11), which declined sharply at 15 cm of white ice and reached a minimum at 40 cm (r = -0.28, Figure 5f).

3.6 Under-ice chlorophyll *a* concentration

The random forest model for under-ice chlorophyll *a* concentration explained 37% of the variation based on a combination of lake morphology, chemistry, ice and snow, but not meteorology. The three most important variables were total phosphorus, lake area, and watershed area. Under-ice chlorophyll concentrations were highest in the bogs, with lower surface area, and the lakes with high phosphorus concentration (Figure 4f). The model selected four variables related to ice and snow: in situ snow depth, mean white ice thickness, fall snow depth, and ice duration. However, the ice and snow variables were relatively unimportant, ranking between 7th and 10th in importance in an 11-variable model (Figure 4f). Under-ice chlorophyll *a* concentrations generally increased with greater in situ snow depth and white ice thickness (Figure 5g and 5h). However, given the low importance of these variables in the model, and the low R^2 of the model, these results should be interpreted with caution.

CHAPTER FOUR: DISCUSSION

Declining ice duration, thinning white ice, and decreasing snow depths across 11 Wisconsin lakes are associated with deeper under-ice photic depths, colder water temperatures, higher dissolved oxygen saturations, and higher chlorophyll *a* concentrations. To our knowledge, this study is the first to explore long-term relationships between changing ice conditions and under-ice ecological conditions in a diversity of lakes. However, we found complex ecological responses to ice loss. Ice alone was insufficient to explain the responses we observed, and should be considered in concert with meteorological, morphological, chemical, and biological characteristics. Our findings suggest that holistic assessments of lake responses to ice loss are necessary for their conservation.

4.1 Ice and snow are changing in Wisconsin lakes

4.1.1 Ice is rapidly being lost in recent decades

The longest ice records in southern Wisconsin, Lakes Mendota and Monona, illustrate significant losses in ice duration since the Industrial Revolution over a 169 year period (1856-2024). Rates of ice loss for Lake Mendota (1.8 days per decade) and Lake Monona (2.2 days per decade) were slightly faster than the Northern Hemisphere average of 1.7 days per decade (Sharma et al. 2021) Their rates of ice loss were comparable to previous research on the same lakes, depending upon the time period studied. Previous studies on one or both lakes have found ice duration declines of 1.9 days per decade between 1853 and 2010 (Magnuson 2010), 2.1 days per decade between 1856 and 1983 (Robertson et al., 1996), and 2.7 days per decade between 1855 and 2004 (Namdar Ghanbari et al., 2009).

Time-series focusing on shorter temporal scales suggest quicker ice loss. For example, in Lakes Monona and Mendota, rates of ice loss were five times faster than our study (9.2 days per decade) between 1968 and 1988 (Anderson et al., 1996). A study on the northern Wisconsin lakes found ice declines of 3.7 days per decade between 1981 and 2015 (Hewitt et al., 2018), more than double our estimated rate of 1.2 days per decade. However, none of the trends in either of these studies were significant (Anderson et al., 1996, Hewitt et al., 2018). Similarly, in our study, the northern Wisconsin lakes showed unanimous but insignificant declines in ice duration. While the World Meteorological Organization recommends a minimum 30-year time-series to study climate change (World Meteorological Organization, 2023), many ice phenology studies use much longer minimum time-series lengths of 100-200 years (i.e., Benson et al., 2012; Magnuson et al., 2000; Sharma et al., 2021). In a comparison of trends for time-series of varying lengths, Hodgkins (2013) found that time series shorter than 50 years do not reliably represent ice phenology trends, which may explain the lack of any significant trends within the 7 northern Wisconsin lakes in our study (42 years of data) and other studies in Wisconsin (Anderson et al. 1996; Hewitt et al., 2018). Nonetheless, winters across Wisconsin are warming (United States EPA, 2016) and lakes are losing ice cover.

4.1.2 Divergent snowfall trends are a main driver of variable ice quality

While ice duration declined in all lakes, trends in ice thickness and snow depth diverged in the northern and southern lakes. Only Trout Bog and Lake Wingra experienced significant changes in ice quality. Trout Bog, in northern Wisconsin, lost 2 cm of black ice per decade and gained 2 cm of white ice per decade. In contrast, white ice thickness declined in Lake Wingra, in southern Wisconsin, by 4 cm per decade. Changes in snowfall likely drive these divergent trends in ice quality, as snow promotes the formation of white ice and slows black ice thickening

(Leppäranta 2015). Within our study, snow depth unanimously increased in the northern lakes (average 1.1 cm per decade) and declined in the southern lakes (average of 1.4 cm per decade). Northern Wisconsin is one of the few areas of the continental United States experiencing greater snowfall under climate change (Cohen et al., 2024), perhaps as a result of increasing lake effect snow (Kunkel et al., 2009). By contrast, snow depth is declining in southern Wisconsin (Cohen et al., 2024). Increasing snow in the northern lakes may have promoted white ice formation on Trout Bog, while thinner snow reduced white ice formation on Lake Wingra. In addition, deeper snow prolongs ice duration (Vavrus et al., 1996). This may explain the insignificant ice duration trends in the northern lakes. While air temperatures in the northern lakes have warmed in the past century (United States EPA, 2016), deeper snow in northern Wisconsin could insulate lakes from ice loss, making trends more difficult to detect. Our results show that snow depth plays a critical role in influencing ice duration and ice quality, and should be explicitly considered in future studies.

4.2 Changing ice influences under-ice ecological conditions

4.2.1 Under-ice photic depths are deeper with less snow and ice

Shorter ice duration and less snow were associated with deeper photic depths over the past 40 years in the studied Wisconsin lakes. Snow has a very high albedo (Bolsenga 1981), so even a small reduction in snow depth can drive much deeper photic depths. In our study, lakes with less snow also had thinner white ice ($r = 0.71$), which permitted greater light transmission through the ice sheet. While snow depth was the more important predictor of photic depth, ice duration plays a key role in snow accumulation and white ice formation. A shorter ice-on season allows less time for snow to accumulate on the ice sheet (Higgins et al., 2021), but this relationship can depend on lake morphology, as indicated by our model. In large lakes, wind often clears snow

from the ice surface, resulting in more black ice formation (Sturm & Liston, 2003) and allowing for greater light transmission (Bolsenga & Vanderploeg 1992). As a result, changing ice duration may have little effect on photic depth in large lakes. However, in small or sheltered lakes, wind does not clear snow effectively (Higgins et al., 2021), leading to darker conditions under ice (Jakkila et al., 2009). For these lakes, shorter ice seasons could have a greater impact, as reduced ice cover limits the dark winter period under ice.

We found that lower chlorophyll *a* concentrations resulted in deeper photic depths. In open water systems, high chlorophyll *a* concentrations can decrease photic depths during events such as algal blooms (ex. Zhang et al., 2022). However, this effect has not been explored in ice-covered lakes, despite documented algal blooms under ice (Reinl et al., 2023; Lenard 2015; Twiss et al., 2012). The ice itself may play a unique role in this phenomenon by limiting light availability. In low light conditions under ice, motile phytoplankton will move toward the ice-water interface, where light is highest (Kelley 1997; Lenard 2015). This can create dense layers of phytoplankton near the ice surface, which has been seen to limit photic depths in marine systems (Ardyna et al., 2020). Lake ice also supports ice-associated algal communities, which attach to the ice sheet at the ice-water interface (Bondarenko et al., 2012; Twiss et al., 2012). In an under-ice photic zone already limited by ice and snow, the combination of ice-associated algae and high phytoplankton densities may reduce photic depths more than previously anticipated.

4.2.2 Under-ice water temperatures are colder with less ice

Later ice-on dates contributed to colder under-ice water temperatures in Wisconsin lakes, both at the lake surface and lake bottom. Ice insulates the water column from heat loss and stabilizes inverse stratification (Bruesewitz et al., 2014). When ice forms later in the year,

surface waters are exposed to the atmosphere for longer. This enables extended heat loss and wind-driven mixing (Yang et al., 2021) and causes cooler water temperatures throughout the water column (Austin 2024). For example, Lake Mohonk, a small, deep lake, had colder under-ice water temperatures when ice-on was delayed (Olesky & Richardson 2024). In our study, under-ice water temperatures were also colder when snow was deeper and white ice was thicker. The high albedo of snow and white ice prevents solar radiation from heating water, resulting in colder under-ice water temperatures (Dombrovsky & Kokhanovsky 2023).

Although bottom and surface under-ice water temperatures responded similarly to changing ice conditions, their meteorological drivers differed. Under-ice bottom water temperatures were primarily influenced by fall meteorology, including fall snow depth, solar radiation, precipitation, and air temperature. While fall is understudied in limnology (Ferrato et al., 2025), recent studies have found that longer fall periods lead to colder under-ice water temperatures (Couture et al., 2015; Ferrato et al., in review). By contrast, under-ice surface water temperatures were more sensitive to meteorological conditions throughout the ice-on season, with seasonal precipitation and snow depth being the most influential meteorological variables in our study. Under-ice surface water can lose heat to the atmosphere as black ice forms (Leppäranta, 2010), and incoming solar radiation can heat surface waters (Dombrovsky et al., 2023). In another study, when snow was removed from a bog lake, surface water temperatures increased in late winter and became more sensitive to meteorological conditions (Dugan et al., 2025). This relationship was more pronounced under high black ice conditions (Dugan et al., 2025). Our findings suggest that under-ice surface water temperatures may be more sensitive to warming or heat loss throughout the winter, whereas under-ice bottom water conditions are primarily driven by fall meteorological conditions and remain more stable through the winter.

4.2.3 Under-ice dissolved oxygen saturations are higher with less ice and snow

Over the past 40 years in Wisconsin, later ice-on dates and less snow contributed to higher dissolved oxygen saturation in lakes under ice. The fall turnover period is critical for increasing oxygen in lakes prior to ice-on (Boegman et al., 2012), with shorter turnover periods causing greater hypoxia and anoxia in lakes (Jane et al., 2022). Later ice onset may cause the water column to become more saturated with dissolved oxygen prior to ice-on, resulting in higher dissolved oxygen saturation throughout the ice-cover season (Jansen et al., 2025). One study on an alpine lake concluded that later ice-on increased dissolved oxygen saturation before ice-on (Flaim et al., 2020). Shorter ice duration has also been associated with a shorter winter hypoxia period in alpine lakes (Smits et al., 2021). However, the effects of shorter ice duration may be mediated by lake size, as the smaller lakes in our study frequently had lower oxygen saturations. Smaller lakes are becoming increasingly hypoxic under ice as winters shorten, whereas larger lakes are becoming more saturated with oxygen (Jansen et al., 2025). The short fetch of small lakes can limit wind-driven mixing and reoxygenation in fall, which carries over to winter (Jansen et al., 2025). Additionally, we found that later ice-on dates were linked to colder under-ice water temperatures. Since oxygen is more soluble in colder water (Marcé et al., 2024), colder water likely promoted higher dissolved oxygen saturation throughout the ice cover period. Water temperatures may be another mechanism by which large lakes become more oxygenated and small lakes more hypoxic, as we found the coldest water temperatures in large, deep lakes.

The interaction between nutrient concentrations and light availability was important in determining dissolved oxygen saturations in our studied lakes. We found that dissolved oxygen saturation was high in large, eutrophic lakes and additionally high in clear lakes. About 1/3 of

surface dissolved oxygen measurements in our dataset were supersaturated, and most of these supersaturation events occurred in eutrophic and clear lakes. However, high dissolved oxygen saturations depended both on high nutrient concentrations and high light availability. Thinner white ice and less snow were associated with more saturated dissolved oxygen in surface water, perhaps because of the increase in available light for photosynthesis (Huang et al., 2021). In Lake Mendota, a year with clear ice resulted in supersaturated dissolved oxygen, but a year with snow on ice did not (Cavaliere et al., 2021). Snow-clearing experiments have shown that, when snow was cleared, the diel cycle in dissolved oxygen saturation was amplified in shallow lakes (Huang et al., 2021), and stormwater ponds became highly oxygenated (Gorsky et al., 2024). By contrast, the bogs in our dataset had high DOC concentrations and low oxygen saturation under ice, regardless of snow depth. High DOC in lakes can accelerate oxygen depletion under ice (Couture et al., 2015). Recent work has highlighted the critical importance of high nutrient concentrations in predicting under-ice hypoxia, especially in small lakes (Jansen et al., 2025). Thus, while less snow and greater light availability can drive higher dissolved oxygen saturations, a lake's response will additionally depend on nutrient availability.

4.2.4 Under-ice chlorophyll a concentrations appear to be unrelated to ice and snow conditions

Ice and snow had little influence on chlorophyll *a* concentrations in our study, despite their influence on light. Recent literature has suggested that light availability is the most important factor limiting algal growth under the ice (Hrycik & Stockwell 2021). As such, we expected a strong signal from snow and ice variables, which influence light availability under ice. However, we found that high phosphorous concentrations were the most important predictor of high chlorophyll *a* concentrations under ice. While studies exploring nutrient concentrations

under ice are rare, a recent mesocosm experiment showed enhanced photoautotrophic growth when nutrients were added (Knoll et al., 2023). Algal growth was enhanced both when snow was removed and when nutrients were added but snow remained (Knoll et al., 2023), suggesting that nutrients and light availability both limit algal growth under ice. The interaction of ice characteristics and nutrient concentrations may account for discrepancies in chlorophyll *a* responses to ice loss observed in other studies. For example, over an 18-year period in the Heiligensee, ice duration declined and chlorophyll *a* concentrations increased (Adrian et al., 1995). The authors noted that phosphorus concentrations increased over the same period, and that nutrient limitation was not a factor in the system (Adrian et al., 1995). In Neusiedler See, chlorophyll *a* concentrations under ice were closely related to phosphorus concentration, but this relationship broke down when ice duration was longer than 60 days (Dokulil & Herzig 2009). Nutrients explained 77.8% of the variation in chlorophyll *a* concentrations under ice in 28 Chinese lakes, while ice and snow variables had no significant effect (Wen et al., 2020). These studies, and our results, highlight the important influence of nutrient concentrations on under-ice algal growth. We propose that there is no single response of chlorophyll *a* concentration to ice loss that can be generalized across lakes. Rather, nutrient availability in lakes must be considered when trying to determine lake chlorophyll *a* responses to ice loss.

Although we did uncover a relationship between snow depth, white ice thickness, and chlorophyll *a* concentrations, their importance in our model was low. We expected that the greater light availability associated with thinner white ice and snow would contribute to higher chlorophyll *a* concentrations (Socha et al., 2023). Contrary to our expectations, chlorophyll *a* concentrations were lower with less white ice and snow. Since the chlorophyll *a* concentrations in our work are photic zone averages, the higher concentrations associated with deeper snow and

thicker white ice could be driven by organisms crowding into a very narrow photic zone (Kelley 1997; Lenard 2015). Motile phytoplankton can even travel into the ice in pursuit of light, surviving inside water pockets in the ice sheet (Barrett et al., 2025). Under-ice phytoplankton communities are often adapted to low light conditions (Bramburger et al., 2023). For example, some phytoplankton species can increase their intracellular chlorophyll concentrations to acclimate to lower light availability (Kirk 2011). In the brighter conditions associated with less snow and thinner white ice, this may not be necessary, leading to lower chlorophyll *a* concentrations in the water column. Adaptation of under-ice phytoplankton to low light may be so extensive that high light availability causes photoinhibition or photobleaching (Bramburger et al., 2023). Additionally, grazing pressure on phytoplankton may be higher as light availability increases, particularly since zooplankton graze selectively on certain taxa (Svensson & Stenson, 1991). Chlorophyll *a* concentrations provide a proxy for under-ice productivity, but were difficult to predict in our studied lakes. This may be related to the many mechanisms that phytoplankton have evolved to survive under ice, some of which confound the use of chlorophyll *a* concentrations as a proxy. To predict how phytoplankton in lakes will respond to changing ice conditions, more research is required, particularly research that incorporates functional traits, community composition, and wider ecosystem effects.

CHAPTER FIVE: IMPLICATIONS AND CONCLUSIONS

This study is one of only a few that have synthesized long-term in situ observations to understand the wide-ranging implications of ice loss to lake ecosystems (e.g., Cavaliere et al., 2021; Hampton et al., 2016 and 2024; Jansen et al., 2025). Over a 40-year period in Wisconsin lakes, we found that declining ice cover was associated with brighter, colder, and more oxygenated conditions under the ice. These physical and chemical changes have important implications for winter lake ecosystems across multiple trophic levels.

5.1 Plankton and food webs

Greater light availability, and its effect on productivity, influence both primary producers and consumers (Hampton et al., 2024). Light availability is an important predictor of under-ice algal blooms (Katz et al., 2015; Reinl et al., 2023), and under-ice algal blooms may occur more frequently as lakes become brighter. Additionally, changing ice conditions can influence the relationship between phytoplankton and zooplankton (Hébert et al., 2021). Food availability is the main driver of zooplankton abundance under ice (Shchapov & Ozersky, 2023). The greater food availability associated with later ice-on and greater under-ice productivity can promote overwintering of consumer zooplankton (Hébert et al., 2021). In cases where zooplankton graze visually and selectively, greater light availability can drive greater consumption of highly nutritious prey by zooplankton under ice (Dekvota et al., 2022; Shchapov & Ozersky, 2023). This can cause different phytoplankton groups, particularly cyanobacteria, to dominate the spring algal bloom (Weyhenmeyer et al., 2002), which raises potential toxicity concerns (Merder et al., 2023). Shifts in entire food webs have already been documented as result of ice loss (Dekvota et al., 2022; Wollrab et al., 2021). More frequent monitoring of intra-seasonal plankton dynamics,

particularly over several subsequent winters, will provide insight into patterns of change in biological systems.

5.2 Fish and biodiversity

The loss of coldwater fish biodiversity in temperate lakes is of increasing concern under climate warming (Roberts et al., 2024). Contrary to our expectations, the increase in colder, well-oxygenated water under ice that we observed may favour native coldwater fishes and provide less suitable habitat for non-native warmwater fishes during the winter (Bloomfield et al., 2022). However, the length of the ice cover period is important for maintaining biodiversity (McMeans et al., 2020). The open-water season is becoming longer and warmer as we lose ice duration, favouring warmwater fishes over coldwater fishes (Alofs et al., 2014; Block et al., 2020; Van Zuiden et al. 2016). Under climate warming, the challenge of a shortening ice cover period may override the benefits of improved under-ice water quality for coldwater fishes and native biodiversity (Wu et al., 2023). If lakes are too warm to support coolwater and coldwater fishes at any part of the year, as is predicted to occur in many Wisconsin lakes (Hansen et al., 2017), under-ice water quality may not be relevant. More research into the effect of shortening winters, and how these short winters affect the subsequent seasons, will clarify the pressures that coldwater fishes face from ice loss.

5.3 Future directions and the importance of long-term monitoring

We highlight the importance of long-term networks to furthering our understanding of the impacts of climate change on ice loss and under-ice ecology (Hampton et al., 2018; Magnuson 1990; Turner et al., 2003). The long-term, detailed data on ice quality, snow conditions, and under-ice ecological parameters from the NTL-LTER provided unique and holistic insights into how lake ecology responds to ice conditions. We reiterate the need for supporting additional

long-term research, particularly during winter, the fastest-warming season (Stine et al., 2009). A key finding of our research was that ice and snow conditions alone are often insufficient to explain ecological responses to ice loss across diverse lakes. To create accurate models, we included additional parameters, such as lake morphology and trophic status. Additionally, we found that divergent responses to ice loss in other studies could be explained by the consideration of these other factors. We encourage further research into lake responses to ice loss to explicitly consider other potential drivers, beyond ice alone, to explain observed patterns. Furthermore, increasing the frequency and spatial coverage of winter sampling–particularly early and late in the season, when ice is thin–will advance our understanding of intraseasonal and spatial variability in ice-covered lakes.

In conclusion, this study is among the few to synthesize long-term, in situ observations of under-ice ecology. We find that ice loss fundamentally alters physical, chemical and biological parameters under ice, which can have cascading effects on phytoplankton, food webs, and fish communities. However, despite the importance of ice, we suggest that it is not necessarily a "master variable" that exerts the greatest control on under-ice ecology. Rather, under-ice ecosystems have drivers as complex as their summer counterparts. We advocate for models of under-ice ecology that explore these diverse drivers as an interconnected system. This will provide necessary context for winter limnology as we move toward a general understanding of under-ice ecological responses to ice loss.

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Figures

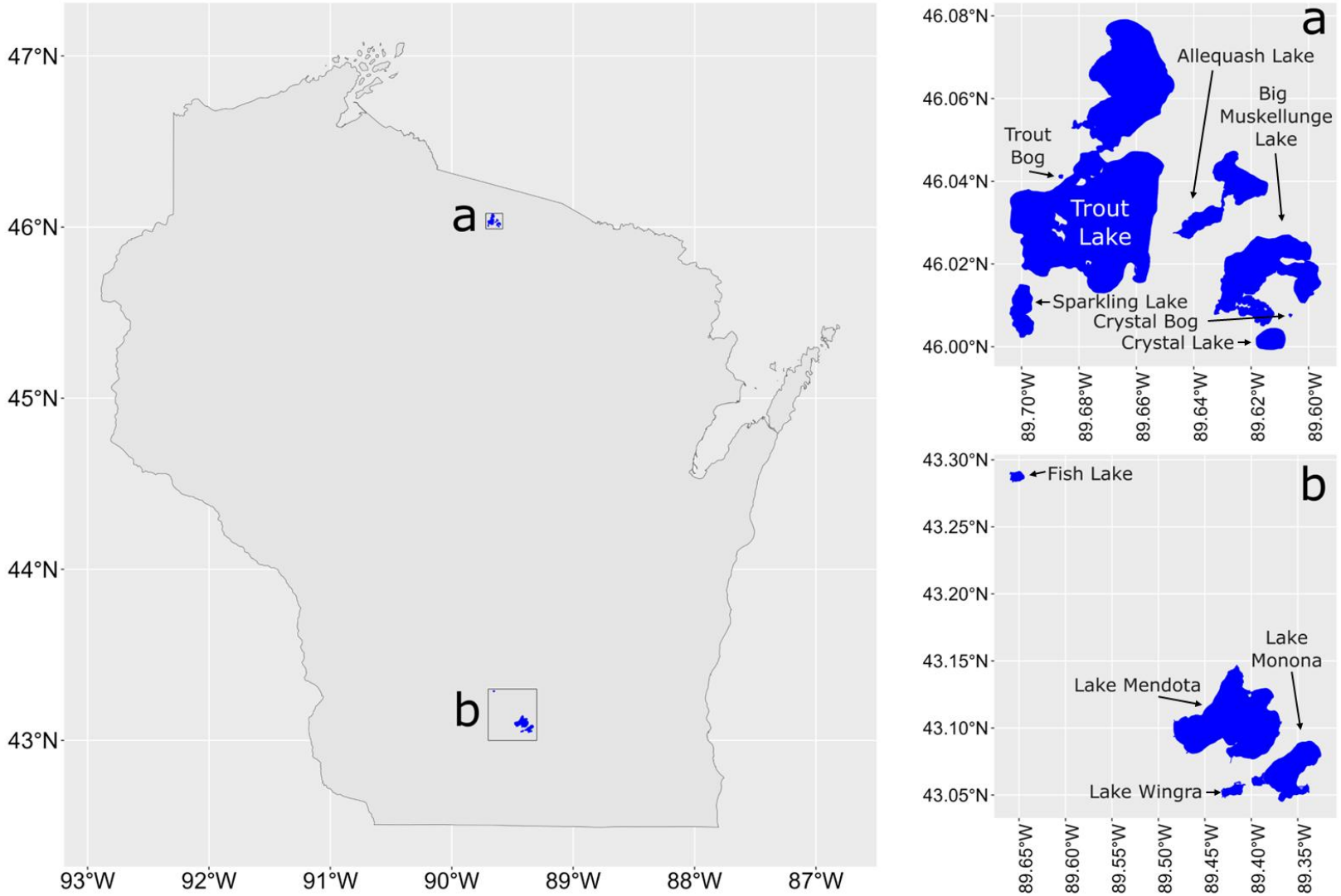

Figure 1. Map of the studied lakes in Wisconsin. Northern lakes are shown in a and southern lakes in b.

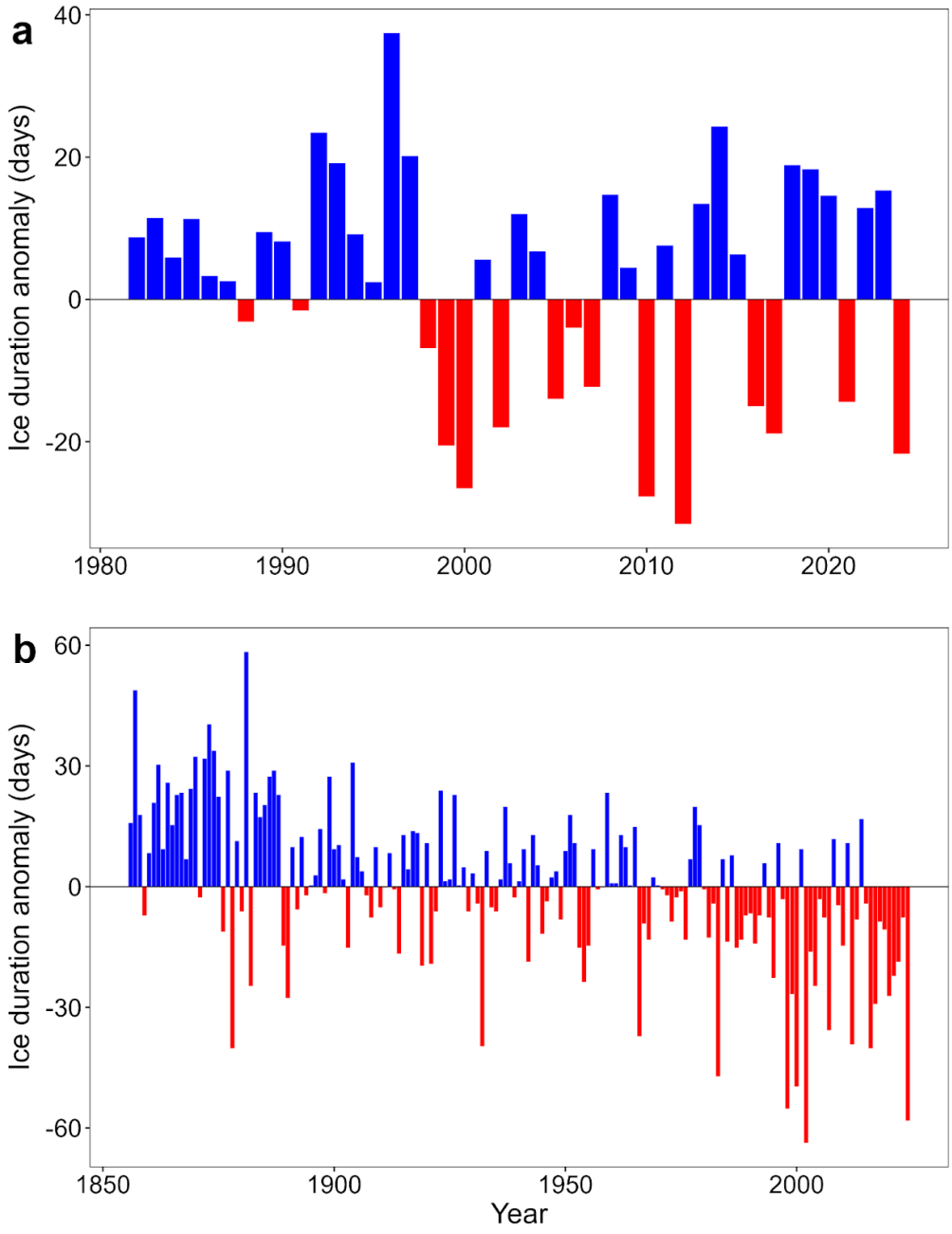

Figure 2. Ice duration anomaly in a) the northern cluster of lakes in Wisconsin and b) Lake Mendota and Lake Monona. We calculated anomalies for each lake by subtracting annual ice duration from average ice duration over the available time period for each lake, then averaging the anomaly of all lakes within a year. The northern lakes had 43 years of continuous ice duration data (1982-2024) and Lake Mendota and Lake Monona had 169 years of continuous ice duration data (1856-2024). Fish Lake and Lake Wingra were excluded from the analysis owing to a relatively short ice record.

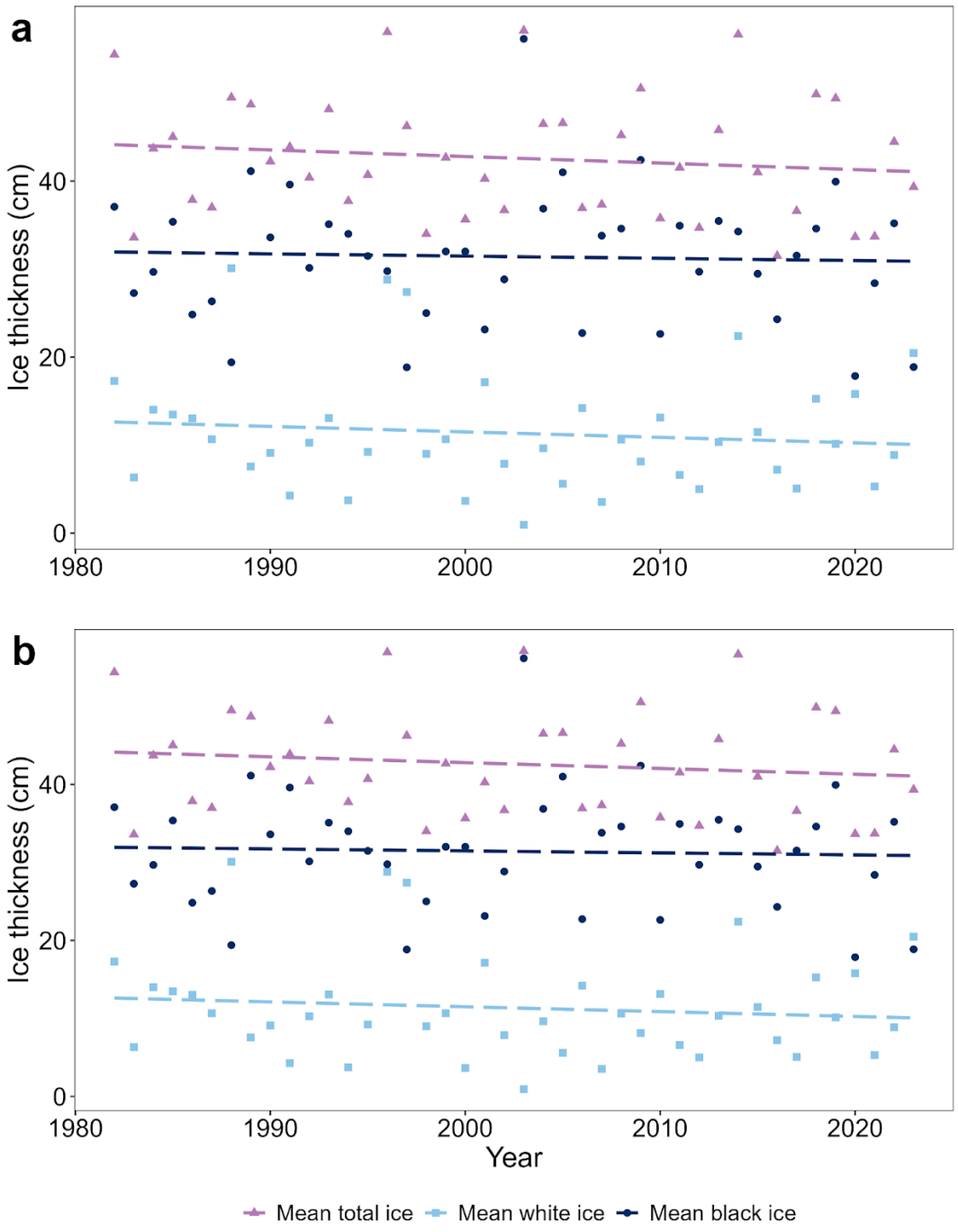

Figure 3. Ice quality changes in cm over time in a) the northern cluster of lakes and b) the southern cluster of lakes over the available time period. We averaged ice thickness for all the lakes in the cluster to produce one point per year. Ice quality data were available in the northern lakes from 1982 to 2023. For the southern lakes, total ice thickness data were available from 1996 to 2023, with black and white ice thickness data available from 2001 to 2023.

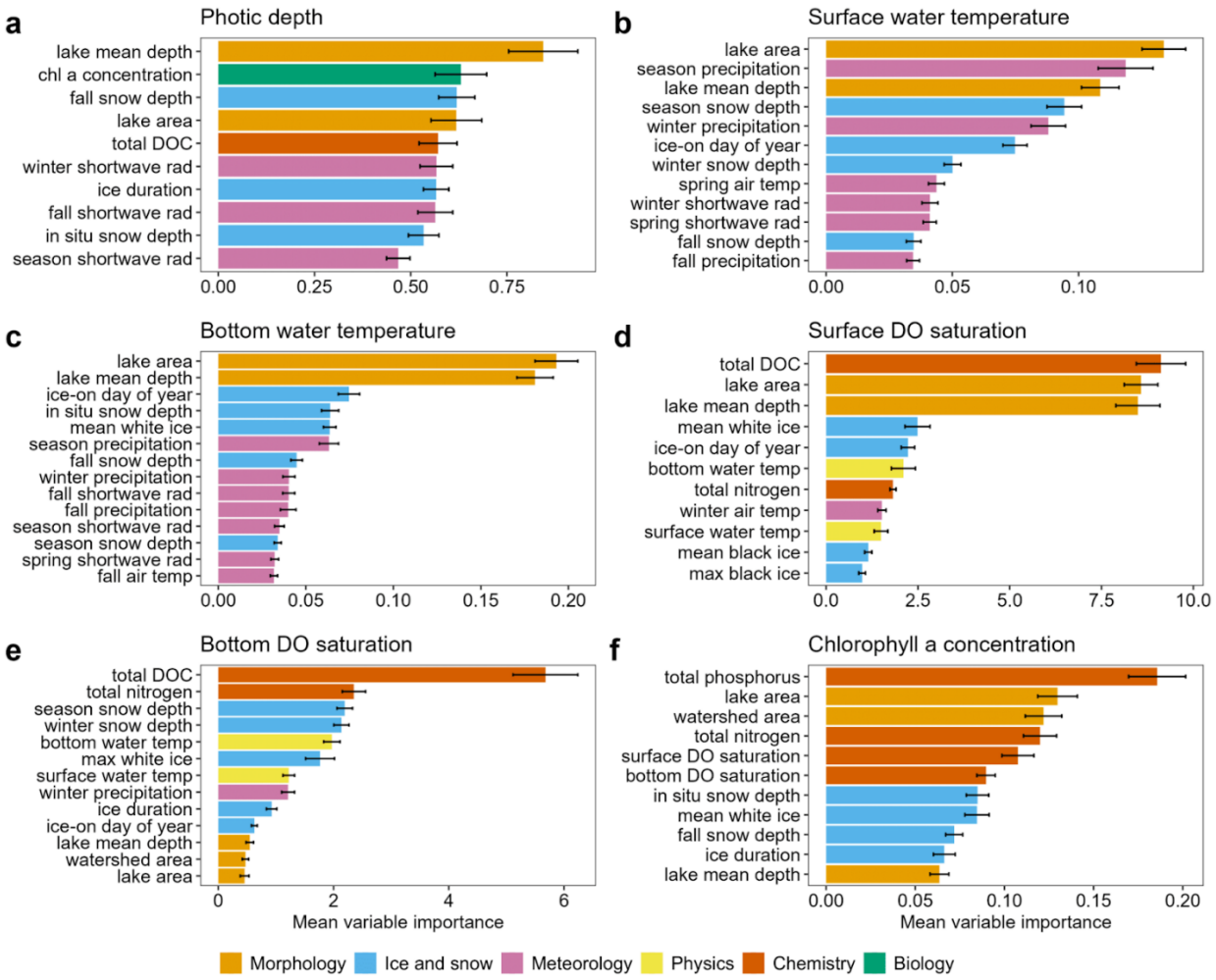

Figure 4. Mean variable importance as measured by root mean square loss of random forest models for: a) photic depth b) surface water temperature c) bottom water temperature d) surface dissolved oxygen saturation e) bottom dissolved oxygen saturation and f) chlorophyll *a* concentration. Bars represent the mean variable importance over 50 permutations, and error bars represent the standard deviation of the variable importance.

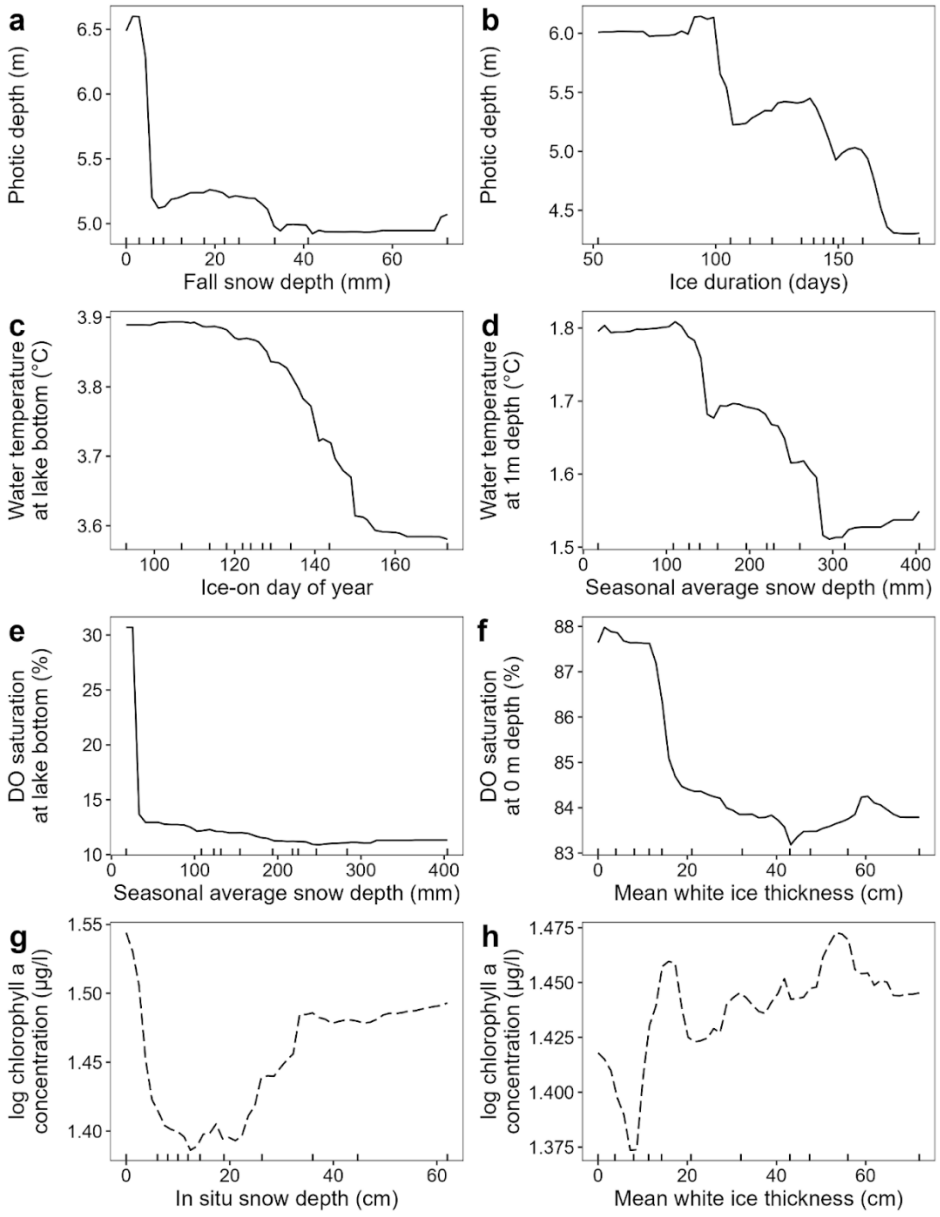

Figure 5. Partial dependence of response variables with key ice and snow variables. The first and second most important ice and snow variables within each model were selected. Plots represent the partial dependence of: a) photic depth on fall snow depth; b) photic depth on ice duration; c) bottom water temperature on ice-on day of year; d) surface water temperature on seasonal average snow depth; e) bottom dissolved oxygen saturation on seasonal average snow depth; f) surface dissolved oxygen saturation on mean white ice thickness; g) log chlorophyll *a* concentration on in situ snow depth; and h) log chlorophyll *a* concentration on mean white ice thickness. Dashed lines represent models with low fit or variables with low importance. Ticks on the x-axis indicate intervals each containing 10% of training observations.

Tables

Table 1. Trends in ice duration, total ice thickness, black ice thickness, white ice thickness, and snow depth for 11 Wisconsin lakes. Trends in ice duration are in days per decade. Trends in ice thickness and snow depth are in cm per decade. Significant trends are indicated with an asterisk (*).

	Lake name	Ice duration	Total ice thickness	Black ice thickness	White ice thickness	Snow depth
Northern lakes	Allequash Lake	-2.0	-0.4	0.2	-0.1	0.7
	Big Muskellunge Lake	-1.1	-1.4	0.7	-1.4	0.8
	Crystal Bog	-1.0	1.5	0.6	0.7	1.1*
	Crystal Lake	-1.0	-0.9	-1.1	0.1	1.6*
	Sparkling Lake	-1.0	-1.4	-2.2	0.3	0.8
	Trout Bog	-0.5	0.3	-2.0*	2.0*	1.5
	Trout Lake	-1.5	0	0.8	-0.6	1.1
	Northern lakes average	-1.1	-0.3	-0.4	0.1	1.1
Southern lakes	Fish Lake	NA	2.2	2.5	-3.1	-2.5
	Lake Mendota	-1.8*	-0.3	-1.0	0.4	-0.4
	Lake Monona	-2.2*	4.6	0	2.0	-2.1
	Lake Wingra	NA	2.3	4.3	-4.1*	-0.6
	Southern lakes average	-2.0	2.2	1.4	-1.2	-1.4

Supplementary Material

Table S1. Names, locations, morphological characteristics, and ice and snow summary statistics of the 11 study lakes. Allequash, Big Muskellunge, Crystal, Sparkling, and Trout Lakes, in addition to Crystal and Trout Bogs, are in northern Wisconsin; Fish Lake and Lakes Mendota, Monona, and Wingra are in southern Wisconsin. SD denotes standard deviation.

Lake name	Latitude	Longitude	Lake area (km^2)	Mean depth (m)	Watershed area (km^2)	Mean ice duration (days)	SD ice duration (days)	Mean ice thickness (cm)	SD ice thickness (cm)	Mean snow depth (cm)	SD snow depth (cm)
Allequash Lake	46.038	-89.621	1.64	2.9	21.51	146.17	16.76	45.01	7.76	12.49	6.26
Big Muskellunge Lake	46.021	-89.612	3.63	7.5	4	140.39	16.53	45	7.9	11.5	5.31
Crystal Bog	46.008	-89.606	0.01	1.7	0.05	151.9	16.4	36.23	6.38	12.16	3.82
Crystal Lake	46.003	-89.612	0.38	10.4	0.86	138.22	15.66	39.09	8.21	11.38	5.53
Sparkling Lake	46.008	-89.701	0.64	10.9	1.4	136.56	16.38	39.38	8.18	35.81	9.95
Trout Bog	46.041	-89.686	0.01	5.6	0.14	154.39	15.01	37.33	6.87	20.9	9.3
Trout Lake	46.029	-89.665	15.65	14.6	46.95	133.41	17.44	42.04	7.48	42.3	9.95
Fish Lake	43.287	-89.652	0.8	6.6	NA	77.08	22.37	33.6	8.36	8.46	9.8
Lake Mendota	43.099	-89.405	39.61	12.8	562	89.13	18.51	33.84	11.85	5.24	5.84
Lake Monona	43.063	-89.361	13.6	8.2	723	90.22	19.26	34.43	8.36	7.4	6.12
Lake Wingra	43.053	-89.425	1.36	2.7	21.2	109.71	17.07	31.6	8.12	6.71	8.37

Table S2. Physical, chemical, and biological summary statistics of the 11 study lakes. Allequash, Big Muskellunge, Crystal, Sparkling, and Trout Lakes, in addition to Crystal and Trout Bogs, are in northern Wisconsin; Fish Lake and Lakes Mendota, Monona, and Wingra are in southern Wisconsin.

	Photic depth (m)			Surface water temperature (°C)			Bottom water temperature (°C)			Surface DO saturation (%)			Bottom DO saturation (%)			Chlorophyll a concentration (µg/L)		
lake name	mean	min	max	mean	min	max	mean	min	max	mean	min	max	mean	min	max	mean	min	max
Allequash Lake	4.15	1	7	1.63	0.50	3.03	4.22	3.00	5.10	74.43	60.63	88.60	10.87	0.00	43.95	3.82	0.00	25.55
Big Muskellunge Lake	6.14	0	15	1.45	0.53	3.17	4.12	2.70	4.83	102.52	85.90	117.65	5.56	0.80	25.63	1.88	0.22	9.72
Crystal Bog	0.94	0	2	2.45	1.47	3.60	3.81	3.07	4.20	34.96	1.73	87.35	4.30	0.00	25.35	21.63	0.30	141.78
Crystal Lake	5.71	1	12	1.87	0.80	3.83	4.25	3.50	4.80	108.48	86.10	126.37	27.57	1.10	65.43	2.43	0.39	12.99
Fish Lake	9.13	2	24	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	7.40	0.30	41.70
Lake Mendota	14.97	5	27	1.11	0.30	2.80	2.62	1.20	4.50	103.23	84.70	142.70	37.02	0.80	130.80	4.17	0.70	14.10
Lake Monona	12.59	6	20	1.47	0.80	2.50	3.39	2.30	4.20	101.63	89.60	134.40	12.90	0.00	69.60	2.88	0.80	8.50
Lake Wingra	3.19	1	7	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	10.14	0.50	31.60
Sparkling Lake	5.38	0	11	1.57	0.50	2.93	4.04	2.70	4.70	99.23	67.93	125.10	7.00	0.80	33.73	4.69	0.28	10.38
Trout Bog	0.52	0	1.5	1.47	0.57	2.20	4.02	2.97	4.30	21.38	0.50	79.67	1.65	0.00	8.03	27.84	1.30	192.30
Trout Lake	5.51	0	14	1.43	0.43	2.90	2.67	1.50	3.65	94.57	84.63	107.53	18.72	1.20	48.60	1.84	0.30	6.30